Response to Reviewer #3

General Comments

This work analyzed the different responses of haze events over the northern and southern NCP during COVID lockdown. The analyses and interpretation were conducted through sensitivity tests of emissions and meteorological fields. It was demonstrated that pervasive emission reduction during COVID lockdown synergistically reduced the PM2.5 pollution in the southern NCP, while it was counteracted by unfavorable meteorological conditions in the northern NCP leading to worse haze events. The methods are sound, and conclusions are important, while the interpretation needs to be strengthened. I recommend minor revisions before publishing at ACP.

Thank you for the positive feedback and constructive suggestions. We appreciate the recognition of the soundness of our methods and the importance of our conclusions. In response, we strengthened the interpretation of our results, enhancing the clarity and depth of the discussion in the manuscript.

We respond to each specific comment in detail below. The reviewers' comments are shown in *black italics*. Our replies are in indented black text, and the modified text is in blue. The annotated line numbers refer to the revised copy of the manuscript.

Specific comments：

Specific.1 *It is argued in the title the work contains insights from six-year simulations, while the main text is heavily based on the analysis for the 3-week (January 21y 21 to February 16ry 16 in the year of 2020) simulation. To avoid exaggerating insights, it is better to remove "Insights from six-year simulation" in the title.*

Thank you for your constructive feedback. The phrase "insights from six-year simulations" in the original title was intended to highlight the climatological averages from 2015 to 2019, which provide a critical baseline for understanding the $PM_{2.5}$ dynamics during the one-month COVID-19 lockdown period. To address this and ensure clarity, we have revised the title and added detailed explanations throughout the manuscript.

[*Title*]:

"Impacts of meteorology and emission reductions on haze pollution during the Lockdown in the North China Plain"

[Lines 189 in *Sect 2.2*]:

In the METEO case, we applied the same emission inventory as the BASE case but with averaged meteorological conditions from 2015 to 2019. These mean meteorological fields were derived by averaging key meteorological variables (**Text S2**).

[*Text S2*]:

Text S2 Mean meteorology from 2015 to 2019

This study's mean meteorology field data was derived by averaging key meteorological variables (e.g., temperature, wind speed, relative humidity, and pressure) from 2015 to 2019. Given that the vertical levels in the NCEP FNL data varied across different years, we did not average the original data directly. Instead, we processed the data using the WRF Preprocessing System (WPS) to ensure consistency. Specifically, we ran WPS yearly to generate the met_em* files containing processed meteorological variables at uniform vertical levels and grid resolution. We then averaged these met_em* files across the six years at each grid point and pressure level, which helped preserve the atmospheric variables' vertical structure and physical coherence. This approach maintained a realistic representation of the atmospheric state by accounting for the multi-year variability while ensuring that the averaged fields were consistent with the WRF-Chem grid resolution. As the WPS processing already matched the data to the model's spatial resolution, no additional interpolation was required, thus ensuring the physical and spatial consistency of the averaged climatological fields used in the WRF-Chem simulations. This multi-year climatological averaging was designed to capture the typical variations in initial and boundary meteorological conditions. This approach provided a robust and representative baseline for multiple years, effectively minimizing the influence of anomalies or extreme weather events characteristic of any individual year.

Specific.2 *There are two observational datasets used in the paper. The location of the IAP sites should be marked in the map as other sites shown in Figure 1.*

Thank you for your helpful suggestion. We revised Figure 1 to include the location of the IAP monitoring site.

[*Figure 1*]:

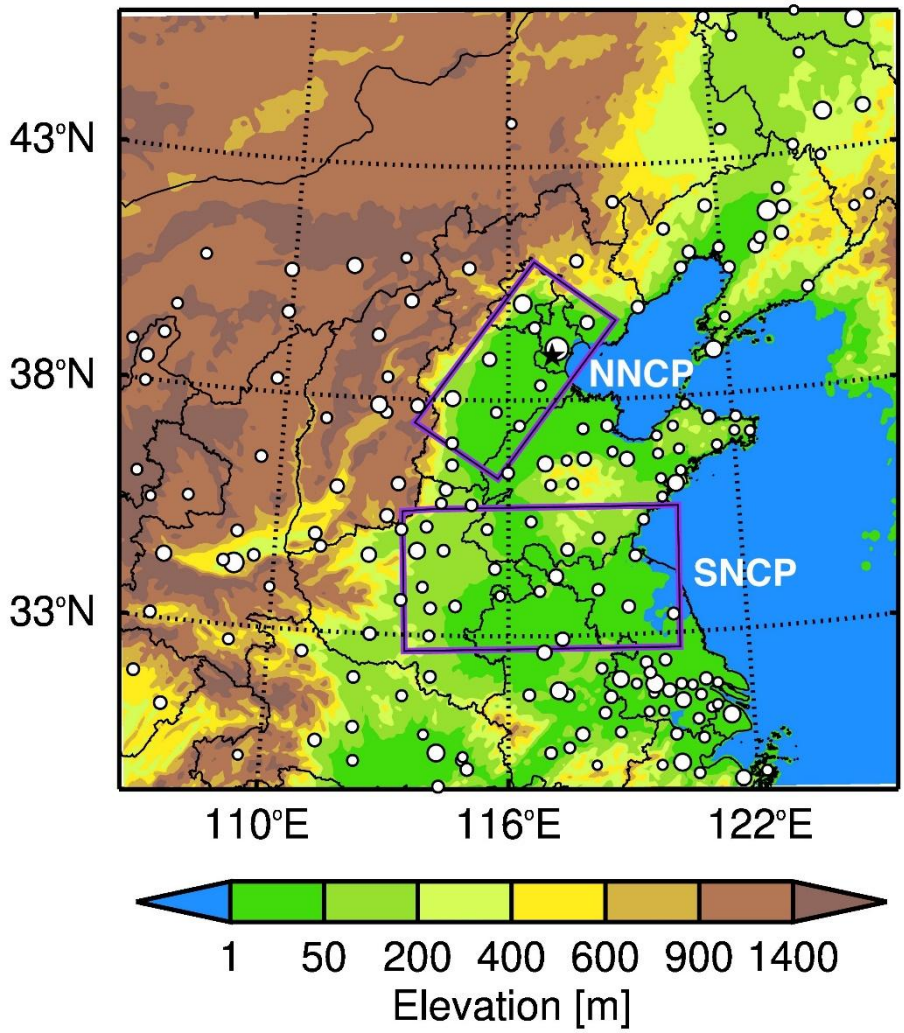

Figure 1. The simulation domain in WRF-Chem, including topography. Circles represent the locations of cities with ambient air quality monitoring sites, with circle size reflecting the number of monitoring sites per city. The IAP observation sites are marked with black pentagons. The regions of interest, NNCP (Northern North China Plain) and SNCP (Southern North China Plain), are highlighted.

[Line 115 in *Sect. 2.1*]:

The second dataset includes chemical compositions such as organic matter, nitrate, sulfate, and ammonium, collected at the Institute of Atmospheric Physics (IAP), Chinese Academy of Sciences in Beijing, China (39°58′28″ N, 116°22′16″ E).

Specific.3 *Line 109, what are the temporal and spatial resolutions of the emission input?*

We added the specific temporal and spatial resolutions of the emission data. Thank you.

[Lines 119 in *Sect. 2.1*]:

We used the Multi-resolution Emission Inventory for China (MEIC), developed by Tsinghua University, with 2016 as the base year (http://meicmodel.org). This emission inventory includes emissions from power plants, transportation, industry, agriculture, and residential activities, with data available at a monthly time scale and a spatial resolution of 6 km. We updated the MEIC inventory to reflect the total provincial emissions estimated for 2020, using near-real-time estimation (Zheng et al., 2021). While the total emissions for each province were updated, the spatial distribution of emissions within each province still followed the intensity proportions from the 2016 MEIC inventory.

Specific.4 *For lines 110-114, it is argued that SNCP has significantly higher emissions than the NNCP, which is not evident from Figure S1. The spatial coverage of SNCP is larger than NNCP. The comparison should be done for region-averaged emission flux per square meter per second. Please show the direct statistical evident of higher emission flux in SNCP than NNCP.*

Thank you for your comment, and we apologize for the confusion caused by the unclear wording. Our intention was not to compare the emissions between SNCP and NNCP directly. Instead, we aimed to highlight that both regions exhibit significantly higher emission levels when compared to the northwestern part of the study area. We have revised the text to clarify this distinction and avoid misinterpretation.

[Lines 132 in *Sect. 2.1*]:

The spatial distribution of primary particles ($PM_{2.5}$) and gaseous pollutants (CO, SO_2, NO_x, NH_3, and HCHO) reveals significantly elevated emission levels across both the NNCP and the SNCP, particularly when compared to the less industrialized northwestern regions of the study area (**Figure S1**). These elevated emissions are primarily driven by dense urbanization and significant industrial activity (Zheng et al., 2021). The topographical features of the NCP, with higher elevations in the north and lower elevations in the south (**Figure 1**), along with substantial pollutant emissions from southern regions, indicate that under persistent southerly winds, pollutants are efficiently transported northward. This northward movement exacerbates air quality degradation, contributing to severe haze episodes in

the NNCP, intensifying regional air quality challenges, and complicating mitigation efforts(Huang et al., 2021).

Specific.5 *For lines 115-118, this is not directly related to dataset description, but rather comments for the topographical characteristics. Consider removing it.*

Removed as suggested. Thank you.

Specific.6 *What is the spin-up time for the WRF-Chem simulation? Please clarify.*

We added the spin-up time to the manuscript. Thank you

[Lines 175 in *Sect. 2.2*]:

For the episode simulations, the spin-up time is 3 days.

Specific.7 *For lines 137-139, besides the initial and boundary meteorological conditions, are the meteorological fields within the spatial domain directly simulated by the WRF-Chem, or is it externally provided by NCEP FNL? If it is directly simulated by the WRF-Chem, please provide more details such as the advection, convection, and boundary layer mixing schemes as the effects of meteorological conditions are a main part of this paper. If it is externally provided by NCEP FNL, please add clarification.*

Thank you for your valuable question. The meteorological fields within the spatial domain are directly simulated by the WRF-Chem model, rather than being externally provided by NCEP FNL. We have added the requested details about the advection, convection, and boundary layer mixing schemes.

[Lines 151 in *Section 2.2*]:

Further details regarding the model settings, initial and lateral meteorological and chemical fields, and anthropogenic and biogenic emission inventory(**Table S1**). We used physical schemes of the WRF single-moment(WSM) 6-class graupel microphysical scheme(Hong and Lim, 2006), the Mellor–Yamada–Janjic (MYJ) turbulent kinetic energy planetary boundary layer scheme (Janić, 2001), the unified Noah land-surface model (Chen and Dudhia, 2001) and the Monin-Obukhov surface layer scheme (Janić, 2001).

[*Table S1*]:

Table S1 Model configuration for the simulation domain, meteorological schemes, chemical mechanisms, initial and lateral conditions, and emission inventories.

Domain	
Size	300 × 300 horizontal grid cells
Center	116°E, 38° N
Horizontal resolution	6 km × 6 km
Vertical resolution	35 vertical levels, uneven intervals, spacing ranging from ~50 m near the surface, ~500 m at 2.5 km above the ground level, and more than 1 km at 14 km above the ground level
Meteorology	
Microphysics scheme	WSM 6-class graupel microphysics scheme (Hong and Lim, 2006)
Boundary layer scheme	MYJ PBL scheme (Janjić, 2002)
Surface layer scheme	Monin-Obukhov surface layer scheme (Janjić, 2002)
Land-surface scheme	Noah land-surface model (Chen and Dudhia, 2001)
Longwave radiation scheme	Goddard (Dudhia, 1989)
Shortwave radiation scheme	Goddard (Dudhia, 1989)
Dry deposition	Wesely (1989)
Wet deposition	CMAQ (Binkowski and Roselle, 2003)
Chemistry	
Gas phase chemistry	SAPRC99 chemical mechanism (Binkowski and Roselle, 2003)
Inorganic aerosols	ISORROPIA version 1.7 (Nenes et al., 1998)
Secondary organic aerosol	Nontraditional VBS parametrization (Li et al., 2011)
Photolysis rates	FTUV radiation transfer model (Tie et al., 2003)
Boundary and initial conditions	
Meteorological	NCEP FNL 6-hr 1° × 1° analysis data
Chemical	CAM-chem 6-hr outputs
Emission inventory	
Anthropogenic	MEIC (Zhang et al. 2009; Li et al., 2017)
Biogenic	MEGAN (Guenther et al., 2006)

Specific.8 *For lines 146-157, how are the scaling factors determined for emission sensitivity test? Please provide rational for the determination of scaling factor. Is it determined relative to certain emissions? Consider adding demonstration that the emissions are back to normal after the scaling? How are the scaling factors applied for emission sensitivity test? Are they applied as a constant for each city?*

Thank you for pointing out the need for further clarification regarding the determination and application of scaling factors in the emission sensitivity test. We have revised the manuscript to provide a more detailed explanation of the emission inventory of the BASE and EMIS cases.

[Lines 119 in *Sect. 2.1*]:

We used the Multi-resolution Emission Inventory for China (MEIC), developed by Tsinghua University, with 2016 as the base year (http://meicmodel.org). This emission inventory includes emissions from power plants, transportation, industry, agriculture, and residential activities, with data available at a monthly time scale and a spatial resolution of 6 km. We updated the MEIC inventory to reflect the total provincial emissions estimated for 2020, using near-real-time estimation (Zheng et al., 2021). While the total emissions for each province were updated, the spatial distribution of emissions within each province still followed the intensity proportions from the 2016 MEIC inventory. Subsequently, we applied a top-down approach to adjust further the emission inventory, iteratively comparing model simulations with observed data to refine the estimates until the simulations closely matched the observations. We validated the final emission inventory using statistical parameters, including normalized mean bias (*NMB*), index of agreement (*IOA*), and correlation coefficient (*r*) (**Text S1**).

[Lines 185 in *Section 2.2*]:

In the EMIS experiment, we used the anthropogenic emission inventory from the BASE case. Still, we excluded any abrupt decreases associated with anthropogenic emission reductions during the COVID-19 lockdown period in 2020, following the provincial emission reduction ratios provided by Huang et al. (2021) (**Table S2**).

[Table S2]:

Table S2 Provincial emission reduction ratios during the COVID-19 lockdown period 2020 in the study area.

Species / Province	CO	NO_x	SO_2	VOCs	$PM_{2.5}$	BC	OC
Beijing	22%	45%	26%	45%	18%	46%	8%
Tianjin	21%	38%	20%	41%	14%	22%	6%
Hebei	15%	45%	16%	36%	12%	17%	5%
Anhui	14%	56%	22%	31%	11%	22%	4%
Inner Mongolia	14%	29%	15%	34%	13%	16%	6%
Shaanxi	19%	45%	18%	34%	13%	22%	5%
Hubei	19%	55%	23%	35%	16%	23%	10%
Jilin	16%	39%	23%	34%	13%	18%	5%
Liaoning	21%	40%	28%	36%	16%	28%	8%
Henan	23%	57%	22%	41%	18%	35%	8%
Shandong	23%	50%	25%	39%	19%	35%	9%
Jiangsu	23%	50%	26%	41%	16%	35%	7%
Shanghai	35%	48%	42%	45%	34%	54%	42%

Specific.9 *Line 187-193, the low-biased sulfate concentrations were attributed to incomplete SO2 oxidation pathway in the WRF-Chem in the paper. But the author showed that SO2 shows great agreements against observations in Figure S3d with NMB 4.8%. If the sulfate underestimation were due to incomplete SO2 oxidation, underestimation of SO2 would be introduced. Please explain.*

Thank you for this insightful comment. While the overall agreement for SO_2 across 65 stations in the NNCP is indeed high (NMB = 4.8%), the nearest monitoring site to the IAP location (within approximately 10 km) shows an SO_2 underestimation with an NMB of -12.1% **(Figure S4)**. This underestimation pattern near the IAP site aligns with the low bias observed in sulfate concentrations at this location. These results suggest that while WRF-Chem captures regional SO_2 concentrations effectively, it may not adequately represent key localized oxidation processes, such as aqueous-phase reactions and heterogeneous transformations, which are crucial for sulfate formation, particularly in urban areas with high emission densities. This could explain why sulfate concentrations at specific locations are underpredicted, even though regional SO_2 levels strongly agree with observations (Liu et al., 2021; Song et al., 2019).

[Lines 221 in *Sect. 3.1*]:

On a regional scale, the model's good performance in SO_2 simulation (NMB = 4.8% in the NNCP) does not entirely explain the sulfate underprediction, particularly near the IAP site, where local SO_2 is underestimated by −12.1% (**Figure S4**). This local discrepancy suggests that WRF-Chem may inadequately capture oxidation processes such as aqueous-phase and metal-catalyzed reactions, leading to sulfate underestimation in urban areas with high pollution levels (Guo et al., 2017; Liu et al., 2017; Zheng et al., 2015). While the model effectively reproduces the temporal variability of critical components, the consistent underestimation of sulfate and ammonium indicates the need for further refinements in the representation of SO_2 emissions and associated oxidation pathways(Cheng et al., 2016; Li et al., 2018).

[*References*]

Guo, H., Liu, J., Froyd, K. D., Roberts, J. M., Veres, P. R., Hayes, P. L., Jimenez, J. L., Nenes, A., and Weber, R. J.: Fine particle pH and gas–particle phase partitioning of inorganic species in Pasadena, California, during the 2010 CalNex campaign, Atmospheric Chem. Phys., 17, 5703–5719, 2017.

Liu, M., Song, Y., Zhou, T., Xu, Z., Yan, C., Zheng, M., Wu, Z., Hu, M., Wu, Y., and Zhu, T.: Fine particle pH during severe haze episodes in northern China, Geophys. Res. Lett., 44, 5213–5221, 2017.

Zheng, G. J., Duan, F. K., Su, H., Ma, Y. L., Cheng, Y., Zheng, B., Zhang, Q., Huang, T., Kimoto, T., Chang, D., Pöschl, U., Cheng, Y. F., and He, K. B.: Exploring the severe winter haze in Beijing: the impact of synoptic weather, regional transport and heterogeneous reactions, Atmospheric Chem. Phys., 15, 2969–2983.

Cheng, Y., Zheng, G., Wei, C., Mu, Q., Zheng, B., Wang, Z., Gao, M., Zhang, Q., He, K., & Carmichael, G. (2016). Reactive nitrogen chemistry in aerosol water as a source of sulfate during haze events in China. Science Advances, 2(12), e1601530.

Li, L., Hoffmann, M. R., and Colussi, A. J.: Role of nitrogen dioxide in the production of sulfate during Chinese haze-aerosol episodes, Environ. Sci. Technol., 52, 2686–2693, 2018.

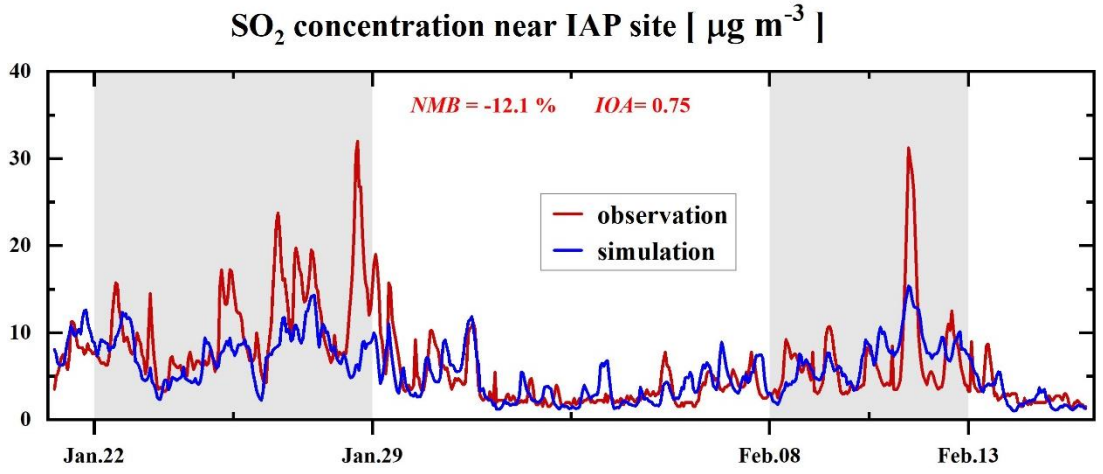

Figure S4. Comparisons of simulated and observed mass concentrations of SO_2 near the IAP site monitoring site from January 21 to February 16, 2020. Blue lines represent simulated concentrations, while red lines indicate observed concentrations.

Specific.10 *For section 3.1, are the simulated concentrations sampled at each site, or each city and then averaged to get regional mean, or directly simulation average for each region? Please clarify.*

Thank you for your question. We clarified that the simulated concentrations were first sampled at each regional observational site.

[Lines 129 in *Sect. 2.2*]:

The simulated concentrations were first sampled at each observational site within the region. These site-specific concentrations were then averaged to calculate the regional mean for the NNCP and SNCP, respectively.

Specific.11 *It is better to show the corresponding scatter plot for each region of NNCP and SNCP as that for all sites in Figure 2.*

Thank you for your suggestion. We have added the corresponding scatter plots for NNCP and SNCP in **Figure S4**, which show statistical comparisons for key pollutants.

[Lines 240 in *Sect. 3.1*]:

The spatial distributions of simulated and observed gaseous pollutants, averaged over the episode, demonstrated strong spatial consistency, with correlation coefficients (r) of 0.67 for O_3, 0.86 for SO_2, and 0.77 for NO_2 across the research domain (**Figure 2e, 2f**). This high consistency was also observed in the NNCP and SNCP regions (**Figure S5**), with correlation coefficients for $PM_{2.5}$ and O_3 of 0.98 and 0.71 in the NNCP, and 0.94 and 0.67 in the SNCP. Similarly, the correlation coefficients for SO_2 and NO_2 were 0.77 and 0.83 in the NNCP, and 0.89 and 0.82 in the SNCP.

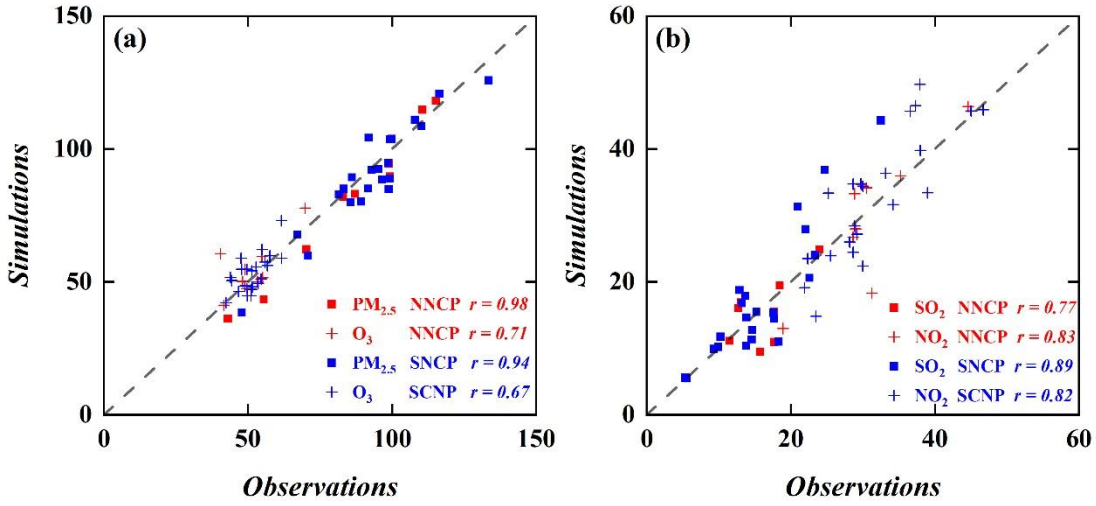

Figure S5. Statistical comparisons of model simulations and observations for (a) $PM_{2.5}$ and O_3, and (b) SO_2 and NO_2 in the NNCP and SNCP regions.

Specific.12 *Figure 2 caption indicates simulated wind fields which are not shown in Figure 2.*

We revised the caption for Figure 2. Thank you.

Specific.13 *Line 228, please add the definition of haze events. Is the criterion of 100 μg/m3 PM2.5 used?*

Thank you for the comment. We included the specific dates for each episode in the manuscript.

[Lines 261 in *Sect. 3.2*]:

Despite the significant reduction in anthropogenic emissions and lower concentrations of NO_2 and SO_2, two unexpected heavy haze episodes occurred in the NNCP. Here, we defined haze events as periods when the daily average $PM_{2.5}$ concentration in the NNCP exceeds 100 μg m^{-3}.

Specific.14 *Line 268, the statement of PM2.5 levels of -50 μg/m3 is confusing as concentration will never be negative. Consider clarifying that it is the effects of meteorological fields on the PM2.5 concentration difference.*

Thank you for the suggestion. The negative value refers to the decrease in $PM_{2.5}$ concentrations due to the influence of meteorological fields. We clarified this in the text.

[Lines 304 in *Sect 3.3*]

Meteorological factors significantly influenced $PM_{2.5}$ concentrations during the study period, as illustrated by the pattern comparisons between the "BASE" and "METEO" simulations (**Figure 5a**). Changes in $PM_{2.5}$ concentrations ranged from decreases of up to 50 μg m^{-3} to increases exceeding 100 μg m^{-3}, revealing an apparent north-south disparity.

Specific.15 *Line 292-294, the statement of regional transport of PM2.5 from SNCP to NNCP does not have strong evidence. There is no prior PM2.5 pollution outbreak in advance in SNCP showed in Figure 3 before EP2 pollution in NNCP. Northward winds are not necessarily indicating pollution transport from SNCP to NNCP when SNCP is clean. Direct evidence may be needed by conducting a sensitivity test by eliminating SNCP emissions and evaluate the PM2.5 differences from that with SNCP emissions.*

Thank you for your insightful suggestion. To provide clearer evidence of the regional transport of $PM_{2.5}$ from SNCP to NNCP, we conducted the "SNCP0" simulation by setting SNCP emissions to zero in the BASE scenario. We quantitatively showed the difference in $PM_{2.5}$ concentrations between the "SNCP0" and "BASE" simulations across different periods, highlighting the contribution of SNCP emissions to $PM_{2.5}$ levels in NNCP.

[Lines 181 in *Sect. 2.2*]

To quantify the influence of SNCP emissions on $PM_{2.5}$ concentrations in NNCP, we also performed an additional sensitivity test (SNCP0) by setting SNCP emissions to zero within the BASE scenario.

[Lines 321 in *Sect. 3.3*]

Meanwhile, the comparison between the "SNCP0" simulation (with SNCP emissions set to zero) and the "BASE" case demonstrated a substantial reduction in $PM_{2.5}$ concentrations in the NNCP (**Figure S8**), particularly during EP2. This reduction, ranging from 15 to 30 μg m^{-3} in certain areas of the NNCP (**Figure S8b**), provides direct evidence that SNCP emissions contribute significantly to $PM_{2.5}$

accumulation in the NNCP via northward transport. This finding underscores the importance of regional transport, facilitated by northward winds, in elevating $PM_{2.5}$ concentrations in the NNCP, especially under meteorological conditions that support pollutant movement from south to north.

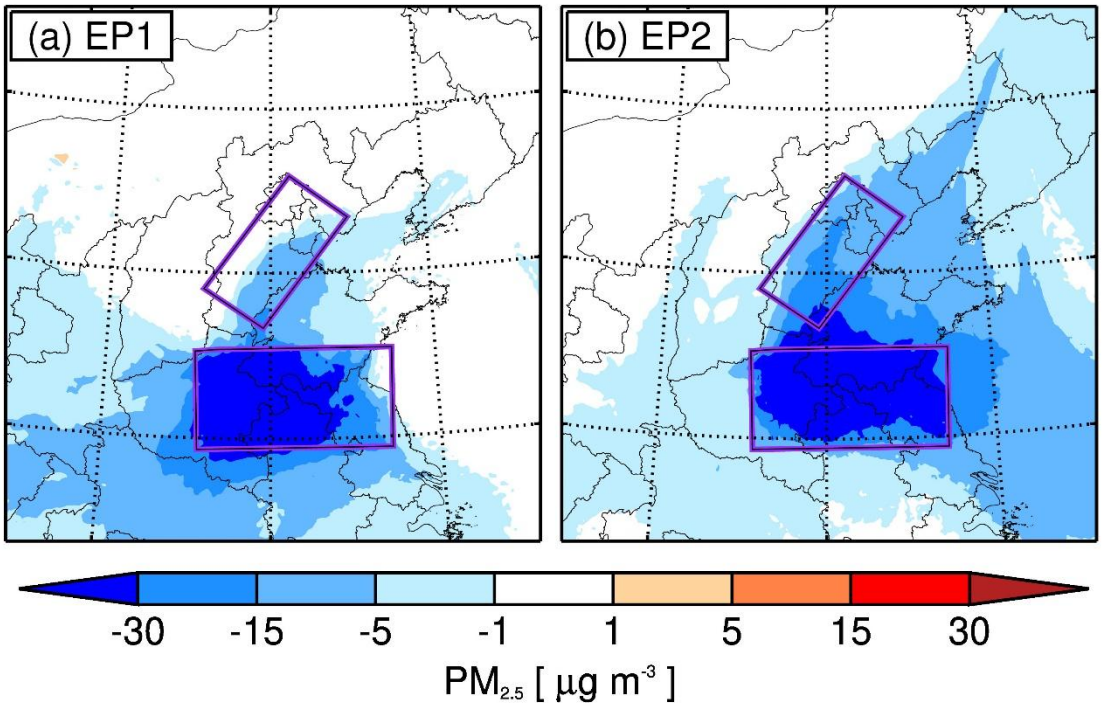

Figure S7. The pattern comparisons of "SNCP0" simulations minus the "BASE" simulation. The color gradient represents $PM_{2.5}$ changes averaged from (a) the EP1 haze period, and (b) the EP2 haze period,

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Specific.16 *Line 301-308, please add more specific evidence of how the increased T2 improves which chemical reaction rates and how higher RH promote particle formation? Is there any direct evidence in this study?*

Thank you for this insightful comment. We have clarified in the revised text that the SEN_METEO simulation captures the influence of elevated temperature (T2) and relative humidity (RH) on secondary aerosol formation through the online WRF-Chem model. Specifically, increased T2 enhances gas-phase oxidation reactions, contributing to secondary organic aerosol (SOA) and nitrate formation, while elevated RH fosters aqueous-phase chemistry that promotes sulfate formation on particle surfaces. Our online WRF-Chem model integrates the effects of T2 and RH directly into the $PM_{2.5}$ concentration results, providing evidence that meteorological factors remained influential in sustaining elevated $PM_{2.5}$ levels during haze episodes, despite

reduced emissions. This highlights the importance of considering meteorological conditions alongside emission reductions in air quality management.

[Lines 345 in *Sect. 3.3*]:

Regional variations in haze episodes underscore the critical role of elevated near-surface temperature (T2) and relative humidity (RH) in driving secondary aerosol formation (**Figure S9**). In the NNCP, elevated T2 accelerates gas-phase oxidation reactions, converting volatile organic compounds (VOCs) and nitrogen oxides (NO_x) into secondary organic aerosols (SOAs) and nitrate aerosols, thus contributing to increased $PM_{2.5}$ levels despite reduced emissions (Huang et al., 2021; Seinfeld and Pandis, 2016). Similarly, elevated RH facilitates aqueous-phase reactions that convert SO_2 into sulfate on particle surfaces, aided by aerosol liquid water, and this effect is particularly pronounced during haze episodes, where high RH accelerates sulfate formation even with decreased emissions (Le et al., 2020; Wang et al., 2020). The online WRF-Chem model captures these interactions in the SEN_METEO simulation, integrating the effects of T2 and RH into the modeled $PM_{2.5}$ concentrations. Although the study does not isolate each specific chemical pathway, the correlation between elevated T2, RH, and higher $PM_{2.5}$ concentrations aligns with previous research, and underscores the pivotal role of meteorological conditions in secondary aerosol formation. This finding highlights the importance of considering meteorological influences in addition to emission reductions, as unfavorable weather conditions can offset the expected improvements from reduced emissions and sustain elevated $PM_{2.5}$ levels. This understanding is essential for developing effective air pollution control strategies that account for emissions and meteorological variability.

[*References*]:

Huang, X., Ding, A., Gao, J., Zheng, B., Zhou, D., Qi, X., ... & He, K. (2021). Enhanced secondary pollution offset reduction of primary emissions during COVID-19 lockdown in China. National Science Review, 8(1), nwaa137.

Seinfeld, J. H., & Pandis, S. N. (2016). Atmospheric Chemistry and Physics: From Air Pollution to Climate Change. John Wiley & Sons.

Le, T., Wang, Y., Liu, L., Yang, J., Yung, Y. L., Li, G., & Seinfeld, J. H. (2020). Unexpected air pollution with marked emission reductions during the COVID-19 outbreak in China. Science, 369(6504), 702-706.

Wang, Y., Yuan, Y., Wang, Q., Liu, C., Zhi, Q., & Cao, J. (2020). Changes in air quality related to the control of coronavirus in China: Implications for traffic and industrial emissions. Science of the Total Environment, 731, 139133.

Specific.17 *Line 319-321, is there any direct evidence that for COVID lockdown period in this study it is also true that it is in a NOx-saturated regime with reduced HOx concentrations? Please add direct evidence in this study.*

Thank you for your comment. We clarified that wintertime ozone production in northern China's urban areas generally occurs in a NOx-saturated regime due to limited HOx radicals and low solar radiation (Seinfeld & Pandis, 2016). During the COVID-19 lockdown, significant NOx reductions reduced ozone titration, allowing ozone concentrations to reach about 65.7 μg/m³, even when $PM_{2.5}$ exceeded 100 μg/m³ (Figure S12). This aligns with findings that reduced NOx can lead to increased ozone levels in NOx-saturated environments, with additional influences from aerosol radiative effects and precursor interactions (Levy et al., 2014; Wu et al., 2020; Le et al., 2020). This supports our conclusion that the NNCP remained NOx-saturated during the lockdown.

[Lines 381 in *Sect 3.4*]:

Wintertime ozone production in urban areas of northern China typically occurs in a NO_x-saturated regime, primarily due to a lack of HOx radicals and limited solar radiation during winter(Seinfeld and Pandis, 2016). Additionally, reduced fresh NO emissions alleviate ozone titration(Levy et al., 2014). Thus, a reduction in NO_x often leads to increased ozone levels. In the NCP during winter, there is usually an inverse relationship between $PM_{2.5}$ and O_3, attributed to the aerosol radiative effect on ozone photochemistry(Li et al., 2017b; Wu et al., 2020). However, during the COVID-19 lockdown, this inverse relationship disappeared in the NNCP, with ozone concentrations reaching approximately 65.7 μg m^{-3} even when $PM_{2.5}$ levels exceeded 100 μg m^{-3} (**Figure S12**). Significant reductions in NO_x emissions reduced ozone titration, resulting in elevated ozone levels despite higher $PM_{2.5}$ concentrations. This pattern aligns with previous findings that in NO_x-saturated environments, reductions in NO_x can increase ozone levels, with additional effects from aerosol radiative influences and precursor interactions shaping the O_3–$PM_{2.5}$ relationship(Le et al., 2020). These dynamics highlight the importance of considering nonlinear chemical and meteorological factors when assessing air quality responses to emission reductions.

[*References*]:

Seinfeld J H, Pandis S N. Atmospheric chemistry and physics: from air pollution to climate change[M]. John Wiley & Sons, 2016.

Levy M, Zhang R, Zheng J, et al. Measurements of nitrous acid (HONO) using ion drift-chemical ionization mass spectrometry during the 2009 SHARP field campaign[J]. Atmospheric environment, 2014, 94: 231-240.

Li G, Bei N, Cao J, et al. Widespread and persistent ozone pollution in eastern China during the non-winter season of 2015: observations and source attributions[J]. Atmospheric Chemistry and Physics, 2017, 17(4): 2759-2774.

Wu J, Bei N, Hu B, et al. Aerosol–photolysis interaction reduces particulate matter during wintertime haze events[J]. Proceedings of the National Academy of Sciences, 2020, 117(18): 9755-9761.

Le, T., Wang, Y., Liu, L., Yang, J., Yung, Y. L., Li, G., and Seinfeld, J. H.: Unexpected air pollution with marked emission reductions during the COVID-19 outbreak in China, Science, 369, 702–706, 2020.

[*Figure S12*]:

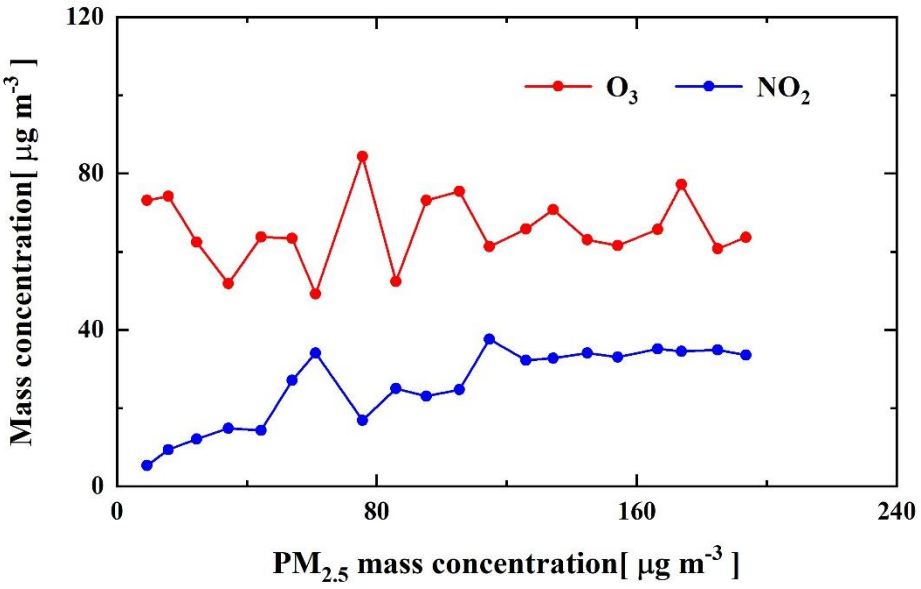

Figure S12. Daytime variation of O_3 and NO_2 (10:00 to 16:00 Beijing Time) as a function of $PM_{2.5}$ concentration during the study period in the NNCP.

Specific.18 *Line 326-327, if the prior argument that NNCP is in a NOx-saturated regime is true, then reduction of NOx does not necessarily lead to a change of O3 concentration.*

Thank you for the comment. In NOx-saturated regimes, reducing NOx emissions generally has a limited impact on O_3 production due to the prevailing chemical conditions where high NOx levels suppress ozone formation through titration. However, during the COVID-19 lockdown, NO emissions reduction alleviated this titration effect, allowing background ozone levels to rise even in a NOx-saturated environment. This dynamic is supported by observations where, despite NOx reductions, ozone concentrations increased, reaching approximately 65.7 μg m^{-3} when $PM_{2.5}$ levels were high (**Figure S12**). This behavior aligns with previous

studies, which found that in NOx-saturated conditions, a decrease in NO emissions can lead to elevated ozone due to reduced titration (Seinfeld & Pandis, 2016; Le et al., 2020; Wu et al., 2020). This highlights the complex and nonlinear relationship between NOx and ozone, which is influenced by chemical and meteorological factors in urban northern China during winter.

[Lines 381 in *Sect 3.4*]:

Wintertime ozone production in urban areas of northern China typically occurs in a NO_x-saturated regime, primarily due to a lack of HOx radicals and limited solar radiation during winter(Seinfeld and Pandis, 2016). Additionally, reduced fresh NO emissions alleviate ozone titration(Levy et al., 2014). Thus, a reduction in NO_x often leads to increased ozone levels. In the NCP during winter, there is usually an inverse relationship between $PM_{2.5}$ and O_3, attributed to the aerosol radiative effect on ozone photochemistry(Li et al., 2017b; Wu et al., 2020). However, during the COVID-19 lockdown, this inverse relationship disappeared in the NNCP, with ozone concentrations reaching approximately 65.7 μg m^{-3} even when $PM_{2.5}$ levels exceeded 100 μg m^{-3} (**Figure S12**). Significant reductions in NO_x emissions reduced ozone titration, resulting in elevated ozone levels despite higher $PM_{2.5}$ concentrations. This pattern aligns with previous findings that in NO_x-saturated environments, reductions in NO_x can increase ozone levels, with additional effects from aerosol radiative influences and precursor interactions shaping the O_3–$PM_{2.5}$ relationship(Le et al., 2020). These dynamics highlight the importance of considering nonlinear chemical and meteorological factors when assessing air quality responses to emission reductions.

Technical corrections：

Technical.1 *Line 245, replace the bell symbol by bell-shaped.*

Changed as suggested. Thank you.